Thinning and Burning in Dry Coniferous Forests of the Western United States: Effectiveness in Altering Diameter Distributions

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Abstract: Western United States land managers are conducting fuel reduction and forest restoration treatments in forests with altered structural conditions. As part of the National Fire and Fire Surrogate (FFS) study, thinning and burning treatments were evaluated for changing forest structure. Shifts between pretreatment and posttreatment diameter distributions at seven western FFS study sites were determined by assessing live tree diameter frequency distributions and the 10th and 90th percentile and mean diameter. Diameter distributions were based on 31,517 live trees within 76 pretreatment units and 25,061 live trees within 85 posttreatment units. Cross-site comparisons were made using meta-analysis. Values for 10th percentile diameter increased at two sites, values for 90th percentile diameter increased at six sites, and values for mean diameter increased at five sites ($P < 0.05$) after active treatments (thin, burn, or thin + burn) compared with control sites. Across the seven western FFS study sites, the overall effect size of the thin treatment increased for 90th percentile and mean diameter, the overall effect size of the burn treatment increased for 10th percentile, 90th percentile, and mean diameter, and the overall effect size of the thin + burn treatment increased for 10th percentile, 90th percentile, and mean diameter. This work indicates that although each of the active treatments was effective in shifting diameter distributions toward larger trees, no single treatment or entry will probably mitigate nearly a century of fire exclusion and fuel accumulation in dry coniferous forests of the western United States. FOR. SCI. 56(1):46–59.

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In the past two decades, unusually large and catastrophic wildfires across the western United States have heightened public awareness of forest ecosystem dynamics and raised concern for forest health. There is growing consensus that active and focused intervention is essential to shift many stands from their current structure and developmental trajectory to conditions that are healthier, are more resilient to fire, and may incorporate natural disturbance regimes prevalent under pre-Euro-American influences (Brown et al. 2004, Graham et al. 2004). Recent presidential initiatives, such as the National Fire Plan, the 10-year Comprehensive Strategy, and the Healthy Forests Initiative, and legislation, such as the Healthy Forests Restoration Act of 2003 (P.L. 108-148), have directed federal land managers, in collaboration with governors and in consultation with a broad range of stakeholders, to reduce hazardous fuels, increase ecosystem resiliency, and improve landscape conditions so that fire can fulfill its appropriate ecological role and benefit other natural processes (US Department of the Interior and US Department of Agriculture 2006a). Consequently, federal agencies have implemented a large number of fuel reduction and forest restoration projects strategically located in low-elevation dry conifer forests.

Pre-Euro-American low-elevation dry conifer forests of the western United States were fundamentally shaped by frequent low- or mixed-severity disturbances (Agee 1993, Covington and Moore 1994, Taylor and Skinner 1998, Everett et al. 2000, 2007, Stephens and Collins 2004, Youngblood et al. 2004, Hessburg et al. 2005, Arabas et al. 2006) mediated by diverse environmental gradients of topography, soils, and weather. Surface fires, ignited predominantly by lightning during the time of year when moisture content of fine fuels was lowest (Agee 1993, Rorig and Ferguson 1999), controlled regeneration of fire-intolerant species, reduced tree density, consumed litter and down wood, opened the stands to increased sunlight, led to vertical stratification of fuels by eliminating fuel ladders between the forest floor and the overstory canopy, and maintained relatively stable plant associations. Consequently, the structure of these low-elevation dry forests generally consisted of open, predominantly widely spaced and old live trees with medium to large diameters, scattered dead trees, and low herbaceous understory vegetation (Fulé et al. 1997, Harrod et al. 1999, Youngblood et al. 2004, Matthew and Taylor 2007). Diameter distributions of both live and dead trees generally were broad, flattened, and unimodal.

Across the western United States, many low-elevation dry conifer forests now have altered structural conditions, contributing to the increased probability of unnaturally severe and extensive wildfires compared with pre-Euro-American forests under natural disturbance regimes (Stephens 1998). These altered structural conditions include greater mass of down woody debris and continuity of the

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fuels mosaic at landscape scales, greater amounts of young multistoried forests with fire-intolerant conifers in both understory and overstory strata, and increased fuel ladders that contribute to greater flame lengths during fires (Agee 1993, Covington and Moore 1994, Arno et al. 1997, Taylor and Skinner 1998, Hessburg et al. 2005, Stephens and Gill 2005). Today, diameter distributions of both live and dead trees tend to be multimodal and heavily weighted to small trees (Hessburg et al. 2005). These altered fuelbeds and shifts in forest structure and composition generally result from fire exclusion and suppression, livestock grazing, timber harvests, and changes in climate and have led to increased competition among trees for below-ground nutrients, water, and growing space, thereby increasing their susceptibility to bark beetles and other forest insects and diseases (Fettig et al. 2007).

Strategies for reducing fuels and restoring fire-adapted ecosystems include thinning live and dead trees and burning surface fuels to reduce the risk of severe surface and crown fires (Brown et al. 2004). Within the context of ecological processes, these treatments may decrease crown bulk density, reduce continuity of the forest canopy, increase the height to the live crown, increase the proportion of fire-resistant tree species, and promote late-successional structures (Graham et al. 1999, Brown et al. 2004, Agee and Skinner 2005). A low thinning treatment will remove small diameter trees from the lower canopy and decrease the diameter variation within the stand and may improve growth of residual trees, allow recovery of potential mortality, encourage and protect differentiation in vertical and horizontal forest structure, and promote large live and dead structure and late-successional characteristics essential for key wildlife habitat (Graham et al. 1999, Agee and Skinner 2005, Kolb et al. 2007). Burning may reduce horizontal continuity of surface fuels, may convert organically bound nutrients to forms more readily available for plant uptake, and may reduce ladder fuels and thus the continuity between surface and canopy fuels (Stephens and Moghoddas 2005, Agee and Lolley 2006, Youngblood et al. 2008). The relative effect of these different strategies on residual tree diameter distributions is not as well understood.

Increased funding levels between 2001 and 2006, coupled with new management tools and improved coordination of vegetation management programs, have enabled federal and state agencies in the United States to use thinning and burning to remove hazardous fuels on nearly 77,000 km^2, over twice the amount of hazardous fuels treatments compared with the previous 6 years (US Department of the Interior and US Department of Agriculture 2006b). Land managers implementing these fuel reduction and forest restoration treatments often lack an understanding of their ecological effects for at least three reasons. First, responses to treatments often have been assessed only over the short-term or are derived for only a single site. Few studies are built on platforms for which long-term studies with periodic remeasurements occur across a network of sites. Second, treatments often have been designed to restore fuelbed conditions or forest composition and structure to historical conditions without a full documentation of historical conditions. Finally, treatments may not fully incorporate the risks, uncertainties, and benefits of multiple environmental stresses associated with a backdrop of natural and human-caused climate change.

Knowledge of fuel reduction and forest restoration treatment effects on tree size hierarchies (Weiner and Solbrig 1984) or changes in tree diameter distributions informs questions of treatment efficacy in realizing treatment objectives, the time when restoration goals might be met in the future, the degree of residual structural diversity, the ability of stands to effectively function as habitat for various wildlife species, future stand growth distribution, and, finally, the potential to produce a range of wood products. Changes in diameter distribution resulting from treatments may take several forms. When a preponderance of trees from the middle of a diameter distribution is removed by thinning, a balanced unimodal diameter distribution may become flattened or reduced in amplitude only. When a preponderance of small diameter trees is removed by burning, a balanced unimodal diameter distribution may become slightly reduced in amplitude and right-shifted, resulting in a larger average stand diameter. When thinning and burning are used in combination, a balanced unimodal diameter distribution may become reduced in amplitude and right-shifted more so than by application of thinning or burning alone. These three hypothetical diameter distributions impart different implications for future management. Because many dry coniferous forests in the western United States now occur with multimodal diameter distributions resulting from disrupted disturbance regimes, fuel reduction and forest restoration treatment effects on residual tree diameter distributions are not easily predicted.

This work is part of the National Fire and Fire Surrogate (FFS) study (Youngblood at al. 2005, McIver et al. 2008), a long-term, multisite, integrated national network of studies established to document consequences of using fire and fire surrogate treatments for reducing fuels and restoring fire-adapted ecosystems. The FFS study include sites extending from the Cascades Range in Washington to South Florida. Each site represents ecosystems that once experienced frequent, low-severity fire regimes. At each site, a common experimental design was used to facilitate broad comparisons of treatment effects. Details of the network and links to individual sites are available (Fire Research and Management Exchange System 2008). Recent work has focused on short-term changes in diverse variables at individual sites and syntheses across sites. For example, thinning and burning reduced the mass of woody fuels and thus the potential fire behavior and severity at western sites (Stephens et al. 2009). Thinning treatments reduced tree density and basal area and increased average stand diameter, whereas burning treatments created more snags across the entire network (Schwilk et al. 2009). Burning did not affect the amount of carbon stored in vegetation. However, burning alone and burning after thinning reduced the forest floor carbon storage, whereas thinning alone and thinning followed by burning reduced carbon storage in vegetation. Thinning alone also reduced the amount of carbon stored in the forest floor (Boerner et al. 2008). There was substantial variability in taxa-specific responses to the treatments by small mammals, suggesting that adaptive management approaches to wildlife may have potential benefits (Converse et al. 2006). Finally, treatments have little effect on most native perennial shrubs and forbs but may lead to short-term increases in non-native invasive plant species (Metlen and Fiedler 2006, Youngblood et al. 2006, Collins et al. 2007).

The focus of this study was to determine whether FFS study treatments in dry coniferous forests of the western United States were effective in initiating short-term changes in forest structure from existing diameter distributions toward those that might persist in late-successional forests. Three separate metrics (10th percentile, mean, and 90th percentile) and cumulative frequency distributions were calculated to estimate the difference between pretreatment and posttreatment diameter distributions at seven sites. Because each FFS study site represents a full and complete experiment, cross-site comparisons were made using a meta-analysis. The five eastern FFS study sites were excluded from consideration because at some eastern sites fuel reduction and forest restoration treatments were designed to reduce tall shrub or herbaceous layers with little attempt to change tree diameter distributions. The following specific objectives are addressed: characterize shifts in diameter distributions as a result of fuel reduction and forest restoration treatments; identify factors influencing shifts in diameter distribution as a result of fuel reduction and forest restoration treatments; and relate this information to ecological restoration of dry coniferous forests of the western United States.

Methods

Study Areas and Study Design

Detailed descriptions of the seven western FFS study sites (Figure 1) are available elsewhere (McIver et al. 2008). The Northeastern Cascades site (47.36°N, 120.53°W) lies between 650 and 1,150 m elevation in north-central Washington on the Wenatchee National Forest. Pretreatment stands were 80-year-old mixtures of ponderosa pine (*Pinus ponderosa* C. Lawson), Douglas-fir (*Pseudotsuga menziesii* [Mirbel] Franco), and some grand fir (*Abies grandis* [Douglas ex D. Don] Lindl.) and averaged about 32 m^2 ha^{-1} in basal area. The Blue Mountains site (45.63°N, 117.24°W) is

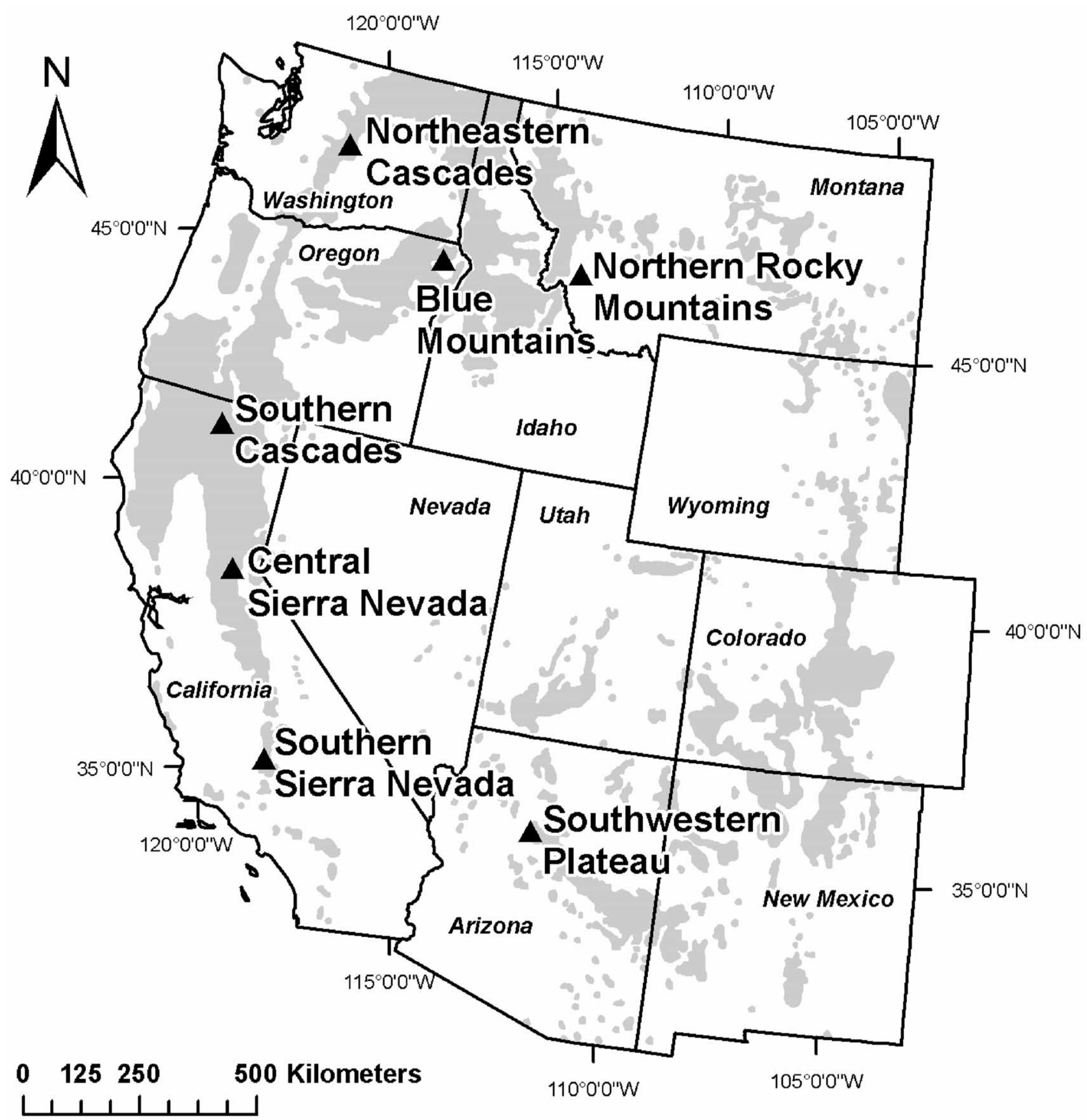

Figure 1. Map of seven western US National FFS study sites, with distribution of *P. ponderosa* shaded.

between 1,040 and 1,480 m elevation in northeastern Oregon on the Wallowa-Whitman National Forest. Pretreatment stands were 70- to 100-year-old even-aged mixtures of ponderosa pine and Douglas-fir and averaged about 18 m^2 ha^{-1} basal area. The Northern Rocky Mountains site (46.89°N, 113.44°W) in western Montana on the University of Montana Lubrecht Experimental Forest is between 1,200 to 1,400 m elevation. Pretreatment stands were dominated by 80- to 90-year-old ponderosa pine and Douglas-fir and averaged about 22 m^2 ha^{-1} basal area. The Southern Cascades site (41.55°N, 121.89°W) in north-central California on the Klamath National Forest lies between 1,480 and 1,780 m elevation. Pretreatment stands of about 90-year-old ponderosa pine and white fir (*Abies concolor* [Gord. & Glend.] Lindl. ex Hildebr.) averaged about 40 m^2 ha^{-1} basal area. The Central Sierra Nevada site (38.91°N, 120.66°W) is between 1,100 and 1,410 m elevation on the University of California Blodgett Experimental Forest in east-central California. Pretreatment stands were approximately 100 years old, and were dominated by sugar pine (*Pinus lambertiana* Dougl.), ponderosa pine, white fir, incense-cedar (*Calocedrus decurrens* [Torr.] Florin), and Douglas-fir and averaged about 50 m^2 ha^{-1} basal area. The Southern Sierra Nevada site (36.53°N, 118.77°W) is between 1,900 and 2,150 m elevation in Sequoia National Park in southern California. Pretreatment stands were multiaged, contained white fir, sugar pine, incense-cedar, ponderosa pine, and some Jeffrey pine (*Pinus jeffreyi* Balf.), and averaged about 60 m^2 ha^{-1} basal area. The final western FFS site was the Southwest Plateau (35.22°N, 112.21°W) in north-central Arizona. This site is between 2,100 and 2,300 m elevation on the Coconino and Kaibab National Forests. Pretreatment stands were about 70 to 90 years old, contained only ponderosa pine, and averaged about 28 m^2 ha^{-1} basal area.

The core experimental design for the FFS study included common treatments, similar treatment replication and plot sizes, and common response variables for all research sites in the network. The four treatments used at six of the seven western FFS sites included (1) thin, (2) burn, (3) thin + burn, and (4) control (Table 1). The thin treatment was a single entry low thin (removing trees from the lower crown classes to favor those in the upper crown classes) conducted with various mechanical cutting and yarding systems. Burns were broadcast underburns conducted after fuels had dried. Treatments at the Southern Sierra Nevada site in Sequoia National Park consisted of both spring and fall burns and control; all burns were considered here as a single treatment. National Park Service policy precluded thinning as a treatment option. The design of active (noncontrol) treatments at each site was guided by a desired future condition or target stand condition uniquely defined for each treatment unit such that 80% of the overstory trees (measured as 80% of the basal area) would survive a wildfire under 80th percentile fire weather condition (Stephens and Moghaddas 2005, Stephens et al. 2009). Treatments were randomly assigned to each unit. Experimental units (treatment units) were whole, discrete stands, or portions of larger stands, all

Table 1. Treatment structure and chronology at seven western Fire and Fire Surrogate (FFS) study sites

	Treatments			
FFS site	Thin	Burn	Thin + burn	Untreated control
Northeastern Cascades	4 units; hand fell, limb, and buck summer 2001; yard with helicopter; slash scattered after thinning	2 units; broadcast burn spring 2004	2 units; hand fell, limb, and buck summer 2001; yard with helicopter; broadcast burn spring 2004	4 units
Blue Mountains	4 units; fell, limb, and buck with mechanical harvester summer 1998; yard with forwarders; slash left in corridors	4 units; broadcast burn fall 2000	4 units; fell, limb, and buck with mechanical harvester summer 1998; yard with forwarders; broadcast burn fall 2000	4 units
Northern Rocky Mountains	3 units; fell, limb, and buck with mechanical harvester winter 2001; yard with forwarders; slash left in corridors	3 units; broadcast burn spring 2002	3 units; fell, limb, and buck with mechanical harvester winter 2001; yard with forwarders; broadcast burn spring 2002	3 units
Southern Cascades	3 units; fell with mechanical harvester summer 1999; whole-tree yard off-unit with skidders	3 units; broadcast burn October 2002	3 units; fell with mechanical harvester summer 1999; yard whole-tree off-unit with skidders; broadcast burn fall 2001	3 units
Central Sierra Nevada	3 units; hand fell, limb, and buck fall 2001; yard with skidders; slash scattered; mechanical mastication of small diameter trees	3 units; broadcast burn fall 2002	3 units; hand fell, limb, and buck fall 2001; yard with skidders; slash scattered; broadcast burn fall 2002	3 units
Southern Sierra Nevada		3 units; broadcast burn fall 2001 3 units; broadcast burn spring 2002		3 units
Southwestern Plateau	3 units; hand fell, limb, and buck fall 2002; yard with skidders; slash scattered	3 units; broadcast burn fall 2003	3 units; hand fell, limb and buck fall 2002; yard with skidders; broadcast burn fall 2003	3 units

having irregular boundaries. Each treatment unit was at least 10 ha and was surrounded by a similarly treated buffer of at least 50 m along each boundary. The number of replicates was consistent across sites ($n = 3$) except at the Blue Mountains site ($n = 4$) by design and at the Northeastern Cascades site ($n = 2$ or 4) where management concerns prevented implementation of some burning treatments after the study was implemented. Pretreatment measurements from nine units were not available at the Southern Cascades site because the national FFS study protocols were overlain on an existing study (Ritchie 2005).

Sampling Procedures

A systematic grid of geo-referenced points 50 m apart and ≥50 m from stand boundaries was established along compass lines within each unit at all western FFS sites. As part of the overall FFS study, some 400 measurement variables were taken across multidisciplinary studies that all referenced grid points. Vegetation was measured at five western sites by using 10 20 × 50 m plots in each treatment unit, with grid points used as referenced corners. Vegetation was measured at the Central Sierra Nevada and Blue Mountains FFS sites by using 21–30 circular 0.04-ha plots that were centered over grid points. Within all plots, a census of live and dead trees by species was conducted before treatment and at the first year after treatment, and tree dbh (1.37 m) was measured. All tree measurements were repeated in 2004 at the Blue Mountains site, in the sixth growing season after the thin treatment and the fourth growing season after the burn treatment. Measurements were also repeated at the Northern Rocky Mountains and Southern Cascades sites in the third growing season after treatments. Data from each western site, arranged as individual tree records, were entered into a single FFS network database and were subjected to multiple error checking, including conformance to experimental variable definitions, consistency in known or expected relationship between multiple measurement variables, and conformance within preestablished measurement variable limits.

Data Analysis

Data for this analysis came from 85 treatment units. For each treatment unit, the national FFS database was queried for diameters of all live trees in pretreatment and the latest posttreatment census. Shifts between pretreatment and posttreatment diameter distributions at seven sites were determined by constructing graphical representations of frequency by diameter classes and by calculating three separate metrics of departure (change in 10th and 90th percentile diameter and change in mean diameter). Live tree lists for each treatment unit at each time period were partitioned into 21 diameter size classes: 0.1–4.9, 5.0–9.9, 10.0–14.9, 15.0–19.9, and so forth to ≥100.0 cm in diameter. Frequency and cumulative frequency histograms were then constructed from these data.

Calculation of 10th and 90th percentile and mean diameter was based on the same data. Although there are an endless number of possible shifts between pretreatment and posttreatment diameter distributions, emphasis here is on the tails of the diameter distribution represented by the 10th and 90th percentile diameter. Mean diameter also was calculated as the best descriptor of overall change in diameter. Assume a pretreatment diameter distribution derived from 150 trees with an approximate modal or normal distribution, with diameters partitioned into 10 uniform classes of 5 cm each (class midpoints range from 2.5 to 47.5 cm), and 10 trees in the smallest and largest classes and 25 trees in the middle class. Next, assume a treatment that reduces the number of trees from 150 to 100 by eliminating trees in the four smallest size classes. The departure or shift in the diameter distribution can be described by the change in 10th percentile diameter (Δ 10th) (22.5 cm posttreatment 10th percentile diameter − 7.5 cm pretreatment 10th percentile diameter = 15 cm), by the change in the 90th percentile diameter (Δ 90th) (47.5 cm posttreatment 90th percentile diameter − 42.5 cm pretreatment 90th percentile diameter = 5 cm), and similarly, by the change in the mean diameter (Δ mean) (31.5 cm posttreatment mean diameter − 24.7 cm pretreatment mean diameter = 6.8 cm). If the total number of trees is reduced the same but the reduction occurs in the middle size classes, Δ mean approaches zero, Δ 10th equals −4.5 cm, and Δ 90th equals 4.5 cm. Finally, assume that the total number of trees is reduced again by one-third and the reduction occurs only in the largest size classes. In this case, Δ 10th equals −4.2 cm, Δ 90th equals −15 cm, and Δ mean equals −7.2 cm. Thus, a positive change value denoted that the posttreatment diameter class distribution was right-shifted compared with the pretreatment diameter size class distribution. A negative value indicated that the posttreatment diameter size class distribution was left-shifted compared with the pretreatment diameter size class distribution. The magnitude of the value, either positive or negative, indicated how far that portion (the 10th or 90th percentile) of the posttreatment diameter size class distribution was shifted compared with the pretreatment diameter size class distribution. Small, positive values of Δ 10th indicate slight declines in the proportion of small diameter trees in the posttreatment diameter distribution, whereas large positive values of Δ 10th indicate treatment elimination of almost all small diameter trees in the posttreatment diameter distribution.

Values of Δ 10th, Δ 90th, and Δ mean as the primary response variable were compared at six of the seven sites with a one-way analysis of variance (ANOVA) of treatment means. All posttreatment units were compared against the mean for the pretreatment control at the Southern Cascades site because of missing pretreatment data (Boerner et al. 2008, Schwilk et al. 2009, Stephens et al. 2009). A paired t test was used for the Southern Sierra Nevada site because only two treatments (burn and control) were conducted. Treatment means and SEs are presented for all sites by treatment. Fine structure differences in changes in fuel loading among treatments were examined by using a priori single degree-of-freedom tests within the context of the ANOVA. The five contrasts were specified as the (1) difference between the control and three active treatments, (2) difference between the mean of the thin and burn treatments and the single thin + burn treatment (thus assessing the interaction effect of the combined treatments), (3) difference between the burn and thin treatment, (4) difference between the burn and the thin + burn treatment, and (5) difference between the thin and thin + burn treatment. A statistical significance level of $P < 0.05$ was used for all univariate tests. Assumptions of normality and equal variances were tested with the Shapiro-Wilk normality test and normal probability plots (Quinn and Keough 2002).

In addition to a site-by-site analysis of the effects of FFS treatments on shifts in diameter distribution, a westwide meta-analysis was also conducted. Typically, the outcome of each study is summarized as an index of effect size, and these indices are summarized across the various studies in a meta-analysis (Gurevitch and Hedges 1999, Rosenberg et al. 2000). Meta-analysis is often conducted when the literature on a particular topic is reviewed with the objective of deriving some overall summary of the conclusions from different studies. Unlike literature-based meta-analyses, work reported here is based solely on the population of seven western FFS study sites. This limitation nullifies one potential weakness of some meta-analyses: the "file-drawer problem" suggests that the strength of literature is strongly biased against publication of nonsignificant results. Effect size is the degree to which a given treatment effect is present in a single study. In this work, effect size was the magnitude of the standardized mean difference between an active treatment and the control. A common approach to calculating effect size is Hedges' g, an estimate of the standardized mean difference that accounts for the fact that the sampling variances for both active treatments and controls are not always equal (Hedges 1981). By using treatment means for 10th and 90th percentile and mean diameters, SDs, and sample sizes, Hedges' effect size g was calculated as

$$g = \frac{\bar{x}_t - \bar{x}_c}{s_p}, \tag{1}$$

where $\bar{x}_t$ is the mean for each active treatment (thin, burn, and thin + burn), $\bar{x}_c$ is the mean for the control, and s_p is the pooled SD for the two groups. Hedges' g calculated in this manner is biased when it is used with small sample sizes (Rosenberg et al. 2000); an adjustment was made to account for this bias. Hedges' d was calculated as

$$d_i = g\left(1 - \frac{3}{4(n_c + n_t - 2) - 1}\right), \tag{2}$$

where d_i is the adjusted effect size for each individual site, n_c and n_t are the number of replicates for the treatment in question at site i, and g is the unadjusted effect size.

To calculate the cumulative effect size at the FFS network scale, a fixed-effects model was used on the assumption that the population of FFS study sites shares a true effect size for the treatments under investigation (Gurevitch and Hedges 1993). Under this model, the test of H_0 that the true effect size was zero was tested by constructing confidence intervals for the true average effect size. Means and 95% confidence intervals of the FFS effect size were calculated using MetaWin 2.0, and effect sizes were considered statistically significant if the 95% confidence interval did not overlap zero (Rosenberg et al. 2000). Finally, a summary analysis was performed to calculate a cumulative effect size, which represented the overall magnitude of the effect across all studies, and the degree of variation within effect sizes. Heterogeneity was the total variation in effect sizes and was similar to the total sum of squares in an ANOVA. Heterogeneity was calculated as

$$q_t = \sum_{i=1}^{n} w_i(d_i - \bar{d})^2, \tag{3}$$

where q_t is the total heterogeneity, n is the number of studies, d is the effect size, and w is the weight for the ith study. The statistic q_t was tested against a χ^2 distribution with $n - 1$ degrees of freedom (Gurevitch and Hedges 1993). The fixed-effect model used study weights defined as the reciprocal of the study variance.

Results

Across the seven western FFS study sites, 31,517 live trees were sampled for diameter within 76 treatment units before treatments were applied (Table 2). Posttreatment live trees totaled 25,061 across 85 units. Values for Δ 10th ranged from −5.8 to 21.7 cm, values for Δ 90th ranged from −5.8 to 33.2 cm, and values for Δ mean ranged from −14.9 to 29.9 cm across all 85 units. Plots of pretreatment and posttreatment live tree frequency and cumulative frequency by diameter size class were generated for each unit and across treatments at each site and compared visually for similarity. Similar patterns were evident.

In thin units, Δ 10th averaged 2.8 ± 1.4 cm, Δ 90th averaged 8.9 ± 1.1 cm, and Δ mean averaged 4.2 ± 0.8 cm, suggesting that posttreatment diameter distributions were right-shifted, and the shift occurred in trees across the lower range of diameters. This shift is exemplified by the cumulative frequency distribution of pretreatment and posttreatment live tree diameters in one unit at the Northern Rocky Mountains site (Figure 2). Trees ≤15 cm in diameter were not included in the census at this site. The number of live trees declined about 61% as a result of treatment, and this decline occurred for trees ≤45 cm in diameter. Most of the difference occurred in the 15.0- to 24.9-cm size class. Cumulative frequency histograms for six sites where thinning occurred depicted the effect of thinning as a departure from the middle of the posttreatment diameter distribution.

In burn units, Δ 10th averaged 3.2 ± 0.7 cm, Δ 90th averaged 5.9 ± 1.6 cm, and Δ mean averaged 3.9 ± 1.8 cm, suggesting that posttreatment diameter distributions were right-shifted, especially for trees ≤45 cm in diameter. This shift is exemplified by the cumulative frequency distribution of pretreatment and posttreatment live tree diameters in one unit at the Northeastern Cascades site, where the number of live trees declined about 19% as a result of treatment, and this decline was generally restricted to trees <30 cm in diameter. Most of the difference occurred in the 10.0- to 14.9-cm size class. Posttreatment cumulative frequency of small live trees was less than pretreatment cumulative frequency, whereas posttreatment large live tree cumulative

Table 2. Dominant tree species, treatment structure, mean and SE for the number of live trees in pretreatment and posttreatment units, and average change in 10th and 90th percentile and mean diameters (cm) at seven western Fire and Fire Surrogate study sites

Site	Tree species	Treatment	No. replicates	Pretreatment tree count	Posttreatment tree count	Δ 10th diameter	Δ 90th diameter	Δ mean diameter
Northeastern Cascades	*P. ponderosa*	Thin	4	377 ± 54	150 ± 34	−1.50 ± 2.10	10.09 ± 1.71	4.99 ± 1.23
	P. menziesii	Burn	2	412 ± 13	353 ± 87	0.69 ± 0.41	3.02 ± 1.49	2.59 ± 1.49
		Thin + burn	2	549 ± 7	195 ± 47	1.18 ± 0.22	11.14 ± 3.78	5.25 ± 2.57
		Control	4	363 ± 42	359 ± 42	0.82 ± 0.21	0.90 ± 0.31	0.97 ± 0.30
Blue Mountains	*P. ponderosa*	Thin	4	697 ± 11	452 ± 73	−1.50 ± 0.88	5.18 ± 0.75	0.57 ± 0.86
	P. menziesii	Burn	4	286 ± 32	247 ± 24	5.11 ± 2.92	3.00 ± 0.55	3.61 ± 0.75
		Thin + burn	4	400 ± 99	145 ± 14	4.27 ± 2.15	7.93 ± 1.26	7.31 ± 1.83
		Control	4	488 ± 35	725 ± 69	−0.69 ± 0.51	−0.17 ± 2.34	−2.42 ± 2.24
Northern Rocky Mountains	*P. ponderosa*	Thin	3	390 ± 36	147 ± 17	3.06 ± 2.08	5.14 ± 1.34	4.85 ± 0.88
	P. menziesii	Burn	3	442 ± 51	382 ± 46	0.38 ± 0.26	1.42 ± 0.05	0.81 ± 0.39
		Thin + burn	3	356 ± 37	115 ± 12	2.03 ± 0.42	4.77 ± 2.37	4.90 ± 0.37
		Control	3	398 ± 27	391 ± 27	0.16 ± 0.08	0.17 ± 0.06	0.13 ± 0.07
Southern Cascades	*P. ponderosa*	Thin	3	N.A.[1]	177 ± 60	13.95 ± 3.92	16.39 ± 1.78	3.59 ± 1.90
	A. concolor	Burn	3	N.A.	293 ± 36	2.73 ± 0.49	0.81 ± 2.33	−11.82 ± 0.94
		Thin + burn	3	N.A.	118 ± 25	13.59 ± 2.03	18.66 ± 1.30	4.42 ± 1.65
		Control	3	362 ± 72	341 ± 51	1.03 ± 0.39	1.04 ± 2.74	−12.76 ± 1.40
Central Sierra Nevada	*P. ponderosa*	Thin	3	489 ± 12	204 ± 13	3.28 ± 0.96	10.59 ± 2.94	8.28 ± 1.26
	P. lambertiana	Burn	3	554 ± 13	328 ± 28	5.42 ± 3.15	5.89 ± 1.28	5.44 ± 1.67
	C. decurrens	Thin + burn	3	531 ± 81	159 ± 40	9.23 ± 2.62	12.48 ± 2.46	12.46 ± 2.77
	P. menziesii	Control	3	859 ± 172	521 ± 33	−0.40 ± 0.21	0.44 ± 0.20	0.07 ± 0.23
Southern Sierra Nevada	*P. ponderosa*	Burn	6	366 ± 20	196 ± 31	3.72 ± 0.87	16.04 ± 3.89	14.30 ± 3.63
	P. lambertiana	Control	3	320 ± 20	313 ± 20	−0.07 ± 0.06	1.21 ± 0.83	0.54 ± 0.65
	A. concolor							
	C. decurrens							
	P. jeffreyi							
Southwestern Plateau	*P. ponderosa*	Thin	3	563 ± 124	189 ± 24	2.40 ± 2.48	6.91 ± 1.21	4.18 ± 2.99
		Burn	3	600 ± 164	561 ± 165	2.46 ± 1.10	0.87 ± 0.57	1.57 ± 0.78
		Thin + burn	3	440 ± 87	140 ± 18	3.37 ± 2.39	11.11 ± 4.71	5.89 ± 1.21
		Control	3	655 ± 144	654 ± 149	0.25 ± 0.09	0.92 ± 0.30	0.52 ± 0.10

[1] N.A., data not available.

frequency was essentially the same as pretreatment large live tree cumulative frequency.

Posttreatment diameter distributions were most strongly right-shifted in thin + burn units, and this shift occurred across more of the diameter distribution compared with that for other treatments. Values for Δ 10th averaged 5.8 ± 1.3 cm, values for Δ 90th averaged 10.8 ± 1.4 cm, and values for Δ mean averaged 6.8 ± 0.9 cm. This shift is exemplified by the cumulative frequency distribution of pretreatment and posttreatment live tree diameters in one unit at the Southwestern Plateau site. The number of live trees declined about 55% as a result of treatment in this unit, and this decline occurred across all size classes for trees <50 cm in diameter. Cumulative frequency histograms generally depicted the effect of thin + burn treatments as a right-shifted departure associated with decreased number of live trees in diameter classes up to and including the mid-diameter classes and little departure associated with the largest diameter size classes.

In control units across the seven western sites, pretreatment and posttreatment diameter distributions were nearly identical. Values for Δ 10th averaged 0.1 ± 0.2 cm, values for Δ 90th averaged 0.6 ± 0.5 cm, and values Δ mean averaged −1.8 ± 1.0 cm. In a few treatment units, the number of small diameter trees increased between the pretreatment and posttreatment censuses (Youngblood et al. 2006).

Values for Δ 10th differed among treatments only at the Southern Cascades and Southern Sierra Nevada sites (Table 3). At the Southern Cascades site, Δ 10th was larger after active treatments compared with controls, was larger for the thin treatment compared with the burn treatment, and was larger for the thin + burn treatment compared with the burn treatment. At the Southern Sierra Nevada site, Δ 10th was larger after the burn treatment compared with the control.

Values for Δ 90th differed among treatments at every site except the Northern Rocky Mountains (Table 4). At the Northeastern Cascades, Blue Mountain, Southern Cascades, and Central Sierra Nevada sites, Δ 90th was larger after active treatments compared with controls. At the Blue Mountains, Southern Cascades, and Southwestern Plateau sites, Δ 90th was larger for the thin + burn treatment compared with the means for both the thin and burn treatments. At both the Northeastern Cascades and Southern Cascades sites, Δ 90th was larger for the thin treatment than for the burn treatment. At the Northeastern Cascades, Blue Mountain, Southern Cascades, and Southwestern Plateau sites and marginally at the Central Sierra Nevada site, Δ 90th was larger for the thin + burn treatment than for the burn treatment. At the Southern Sierra Nevada site, Δ 90th was larger after the burn treatment than after the control treatment.

Values for Δ mean differed among treatments at every site except the Northern Rocky Mountains and Southwestern Plateau (Table 5). At the Blue Mountain, Northern Rocky Mountains, Southern Cascades, and Central Sierra

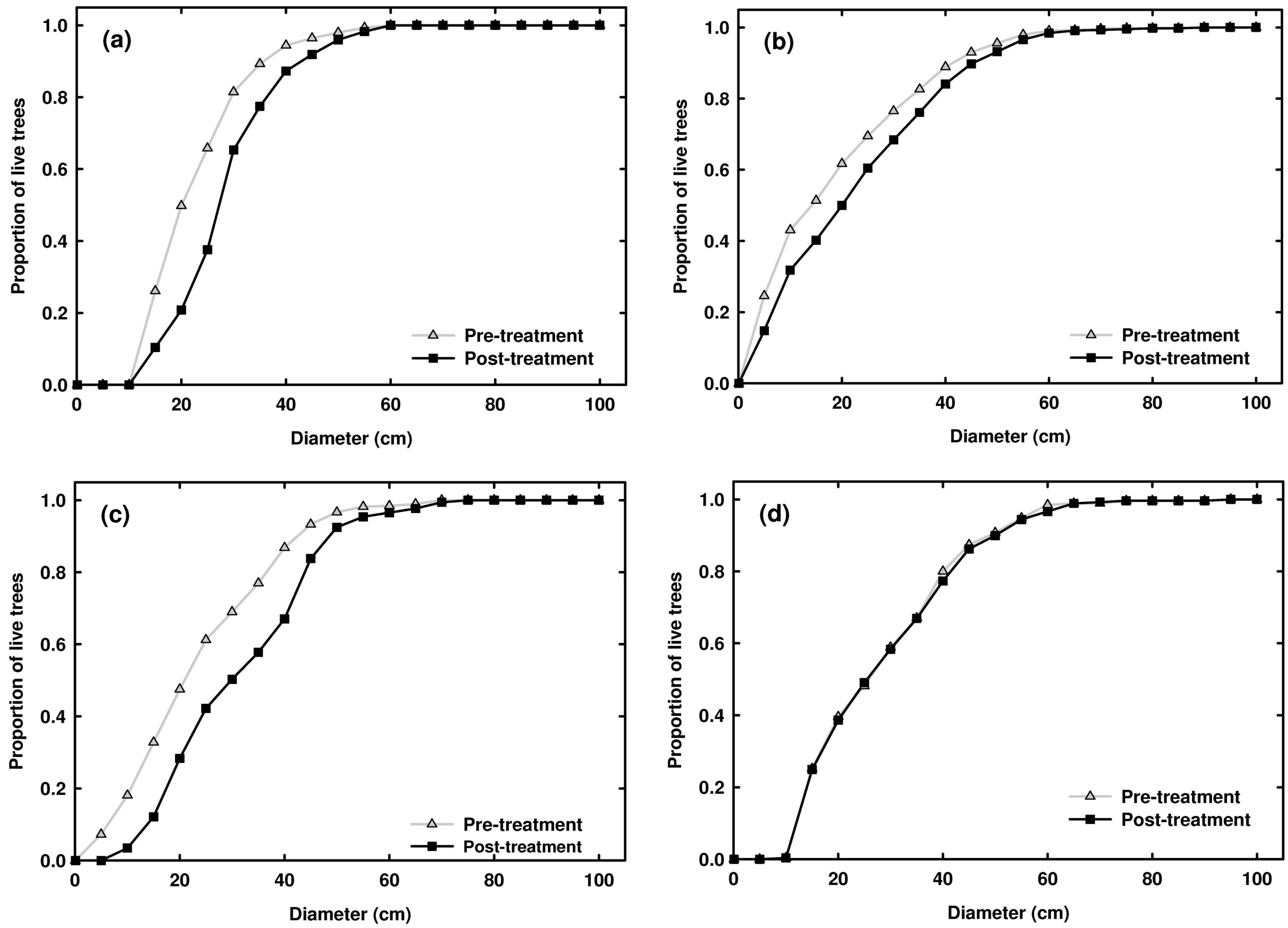

Figure 2. Cumulative frequency distribution of live tree diameter (pretreatment and posttreatment) for four FFS study treatments at representative treatment units, including (a) thin treatment at the Northern Rocky Mountains site, with 448 live trees pretreatment and 173 live trees posttreatment, (b) burn treatment at the Northeastern Cascades site, with 541 live trees pretreatment and 440 live trees posttreatment, (c) thin + burn treatment at the Southwestern Plateau site, with 387 live trees pretreatment and 173 live trees posttreatment, and (d) control treatment at the Southern Cascades site, with 270 live trees pretreatment and 269 live trees posttreatment.

Table 3. Results of ANOVA for the change in 10th percentile diameter after four fuel reduction and forest restoration treatments at six western US Fire and Fire Surrogate study sites and results of a paired *t* test for the Southern Sierra Nevada site

Site	Mean square error	*F* (or *T*)	Contrast	*P*
Northeastern Cascades	5.097	0.69		0.582
Blue Mountains	45.541	3.21		0.062
Northern Rocky Mountains	5.752	1.68		0.248
Southern Cascades	142.968	9.60		0.005
Control versus active			−27.19	0.008
Thin and burn versus thin + burn			−10.50	0.091
Burn versus thin			−11.22	0.007
Burn versus thin + burn			−10.86	0.009
Thin versus thin + burn			−0.36	0.911
Central Sierra Nevada	48.669	3.66		0.063
Southern Sierra Nevada		−2.96		0.021
Southwestern Plateau	5.241	0.53		0.672

Nevada sites, Δ mean was larger after active treatments compared with controls and was larger for the thin + burn treatment compared with the mean of both the thin and burn treatments. At the Northern Rocky Mountains and Southern Cascades sites, Δ mean was larger for the thin treatment than for the burn treatment. At the Northern Rocky Mountains, Southern Cascades, and Central Sierra Nevada sites, Δ mean was larger for the thin + burn treatment than for the burn treatment. At the Blue Mountains site, Δ mean was larger for the thin + burn treatment than for the thin treatment. At the Southern Sierra Nevada site, Δ mean was larger after the burn treatment than for the control treatment.

In addition to the visible differences among cumulative frequency histograms, there were consistent differences

Table 4. Results of ANOVA for the change in 90th percentile diameter after four fuel reduction and forest restoration treatments at six western US Fire and Fire Surrogate study sites and results of a paired *t* test for the Southern Sierra Nevada site

Site	Mean square error	*F* (or *T*)	Contrast	*P*
Northeastern Cascades	80.494	9.32		0.001
Control versus active			−21.54	0.004
Thin and burn versus thin + burn			−9.163	0.097
Burn versus thin			−7.078	0.024
Burn versus thin + burn			−8.12	0.025
Thin versus thin + burn			−1.043	0.693
Blue Mountains	46.980	5.93		0.010
Control versus active			−16.62	0.005
Thin and burn versus thin + burn			−7.69	0.046
Burn versus thin			−2.19	0.294
Burn versus thin + burn			−4.94	0.029
Thin versus thin + burn			−2.75	0.192
Northern Rocky Mountains	18.176	3.26		0.081
Southern Cascades	278.272	20.84		<0.001
Control versus active			−32.76	0.002
Thin and burn versus thin + burn			−20.12	0.005
Burn versus thin			−15.58	<0.001
Burn versus thin + burn			−17.85	<0.001
Thin versus thin + burn			−2.27	0.469
Central Sierra Nevada	86.638	7.04		0.012
Control versus active			−27.63	0.004
Thin and burn versus thin + burn			−8.48	0.126
Burn versus thin			−4.70	0.140
Burn versus thin + burn			−6.59	0.051
Thin versus thin + burn			−1.89	0.528
Southern Sierra Nevada		−2.59		0.036
Southwestern Plateau	74.648	4.14		0.048
Control versus active			−16.12	0.094
Thin and burn versus thin + burn			−14.44	0.043
Burn versus thin			−6.04	0.120
Burn versus thin + burn			−10.24	0.018
Thin versus thin + burn			−4.20	0.260

Table 5. Results of ANOVA for the change in mean diameter after four fuel reduction and forest restoration treatments at six western US Fire and Fire Surrogate study sites and results of a paired *t*-test for the Southern Sierra Nevada site

Site	Mean square error	*F* (or *T*)	Contrast	*P*
Northeastern Cascades	13.916	3.02		0.094
Blue Mountains	69.42	7.19		0.005
Control versus active			−18.73	0.005
Thin and burn versus thin + burn			−10.45	0.018
Burn versus thin			−3.04	0.192
Burn versus thin + burn			−3.71	0.118
Thin versus thin + burn			−6.75	0.010
Northern Rocky Mountains	19.647	24.62		<0.001
Control versus active			−10.16	0.001
Thin and burn versus thin + burn			−4.15	0.011
Burn versus thin			−4.04	0.001
Burn versus thin + burn			−4.10	0.001
Thin versus thin + burn			−0.05	0.944
Southern Cascades	266.473	38.79		<0.001
Control versus active			−34.48	<0.001
Thin and burn versus thin + burn			−17.07	0.002
Burn versus thin			−15.42	<0.001
Burn versus thin + burn			−16.24	<0.001
Thin versus thin + burn			−0.827	0.709
Central Sierra Nevada	81.865	9.00		0.006
Control versus active			−26.13	0.003
Thin and burn versus thin + burn			−11.21	0.030
Burn versus thin			−2.84	0.283
Burn versus thin + burn			−7.02	0.021
Thin versus thin + burn			−4.19	0.128
Southern Sierra Nevada		−2.58		0.036
Southwestern Plateau	17.907	2.16		0.171

among treatments with respect to changes in diameter distributions as measured by Δ 10th, Δ 90th, and Δ mean. Whereas the thin treatment at the Southern Cascades and Central Sierra Nevada sites resulted in a significant effect size, the overall effect size of the thin treatment compared with that of the control treatment on changes in diameter distribution as measured by Δ 10th was not significantly different from zero (mean effect 0.5 ± 0.9) (Figure 3). The Southern Sierra Nevada site was the only site for which the effect size for the burn treatment compared with that for the control treatment was positive and significant for Δ 10th, yet the overall mean effect size of the burn treatment across the seven western FFS study sites was also significant (1.1 ± 0.8). Three FFS study sites had significant positive effect sizes for the thin + burn treatment: Northern Rocky Mountains, Southern Cascades, and Central Sierra Nevada sites. The overall effect size of the thin + burn treatment on changes in diameter distribution as measured by Δ 10th was significantly different from zero (mean effect 1.7 ± 1.0). The overall effect of the burn and thin + burn treatment at western FFS study sites was considered significant because the 95% confidence intervals did not encompass zero.

Across the western FFS study sites, the overall effect size of the thin treatment compared with that of the control treatment on changes in diameter distribution as measured by Δ 90th was 2.4 ± 1.1 (Figure 4). Whereas the burn treatment resulted in a significant positive effect size at only the Northern Rocky Mountains and Central Sierra Nevada sites, the overall effect size of the burn treatment compared with that of the control treatment for Δ 90th was 1.0 ± 0.8. The thin + burn treatment resulted in a significant positive effect size at four sites, and the overall effect size of this treatment compared with that of the control treatment for Δ 90th was 2.1 ± 1.1. Effect sizes for all three treatments were considered significant because their 95% confidence intervals did not encompass zero.

Similarly, the thin treatment resulted in a significant positive effect size for Δ mean at four sites, and the overall effect size of the thin treatment compared with that of the control treatment on changes in diameter distribution as measured by Δ mean was 1.8 ± 1.0 (Figure 5). The overall effect size of the burn treatment compared with that of the control treatment for Δ mean was 1.2 ± 0.8. The thin + burn treatment resulted in significant positive effect sizes at five sites: Blue Mountains, Northern Rocky Mountains, Southern Cascades, Central Sierra Nevada, and Southwestern Plateau. The overall effect size of the thin + burn treatment compared with that of the control treatment for Δ mean was 2.8 ± 1.2. Effect sizes for all three treatments were considered significant because their 95% confidence intervals did not encompass zero.

Discussion

Based on breast-height diameter measurement of nearly 57,000 live trees across seven western FFS study sites, fuel reduction and forest restoration treatments resulted in detectable but only relatively small changes in size hierarchy or diameter distribution. Tree diameter distributions are known to take one of several general forms, including unimodal, multimodal, decreasing or reverse J, or irregular distributions. These common tree diameter distributions develop through differences in competitive status, age, growth rate, and genetic potential, in addition to differences of the

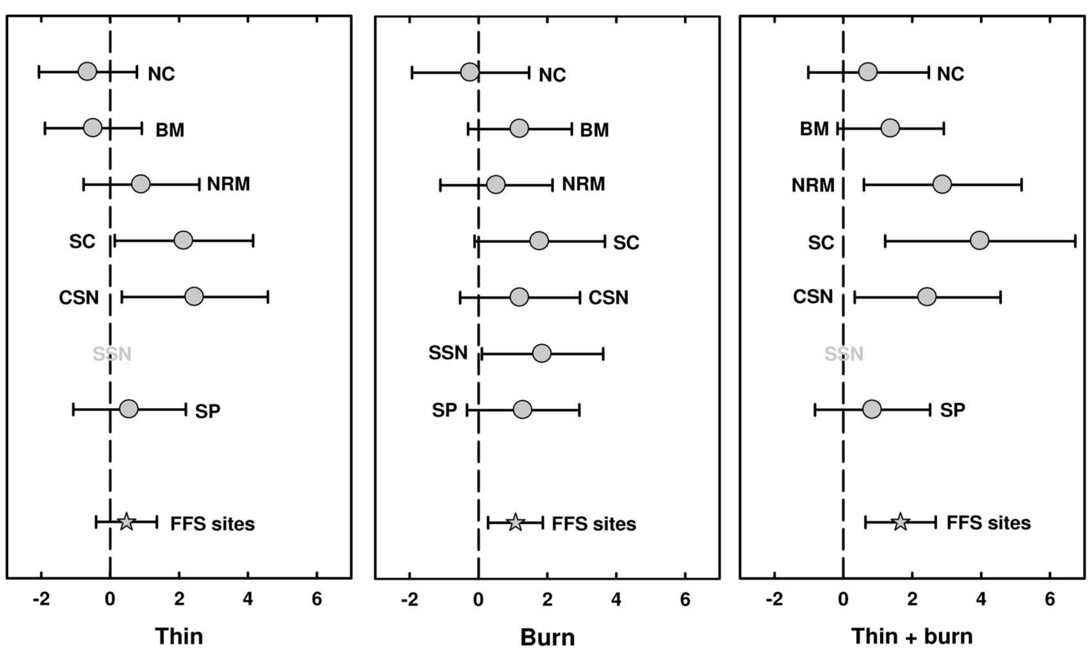

Figure 3. Summaries of the meta-analysis of the effects of three FFS study treatments on changes between pre- and posttreatment 10th percentile diameters (cm). Symbols represent mean effect size, with horizontal lines depicting the 95% confidence interval for the mean effect size. The western FFS network mean is indicated by the star. Sites for which insufficient data were available to test a given treatment are indicated by the site code in gray along the zero effect line. Site codes are as follows: NC, Northeastern Cascades; BM, Blue Mountains; NRM, Northern Rocky Mountains; SC, Southern Cascades; CSN, Central Sierra Nevada; SSN, Southern Sierra Nevada; and SP, Southwestern Plateau.

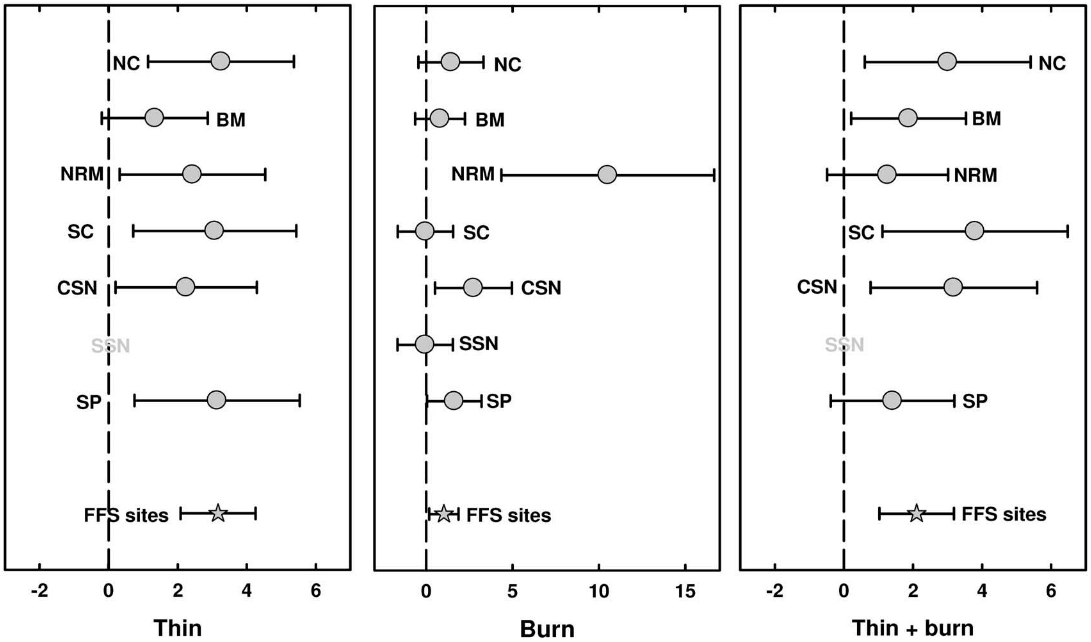

Figure 4. Summaries of the meta-analysis of the effects of three FFS study treatments on changes between pre- and posttreatment 90th percentile diameters (cm). Symbols represent mean effect size, with horizontal lines depicting the 95% confidence interval for the mean effect size. The western FFS network mean is indicated by the star. Sites for which insufficient data were available to test a given treatment are indicated by the site code in gray along the zero effect line. For site codes, see the legend to Figure 3.

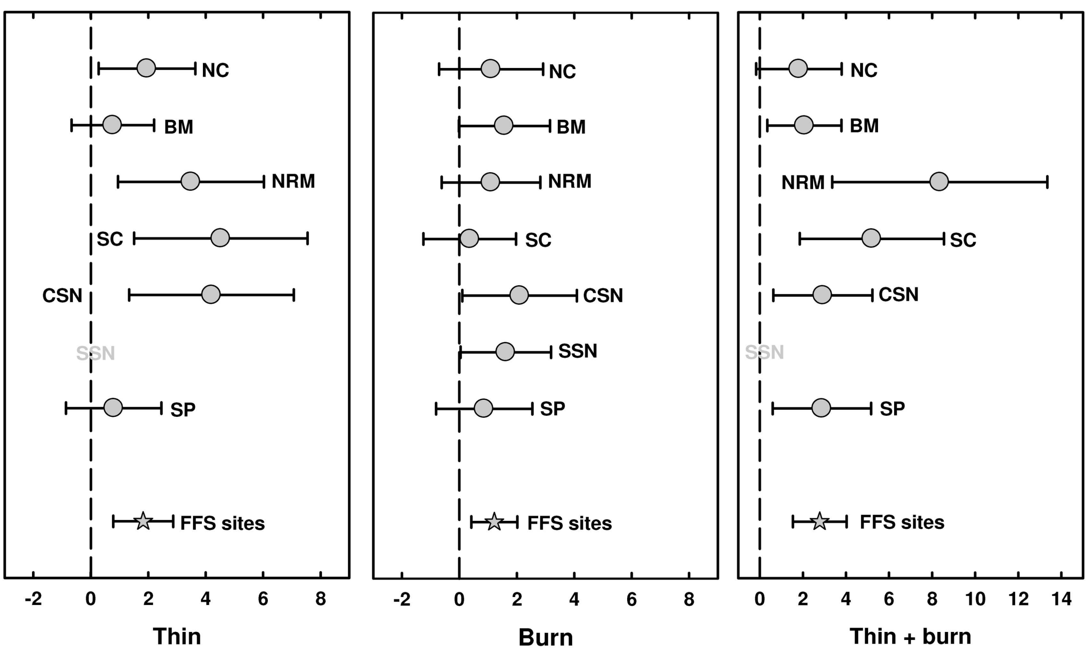

Figure 5. Summaries of the meta-analysis of the effects of three FFS study treatments on changes between pre- and posttreatment mean diameters (cm). Symbols represent mean effect size, with horizontal lines depicting the 95% confidence interval for the mean effect size. The western FFS network mean is indicated by the star. Sites for which insufficient data were available to test a given treatment are indicated by the site code in gray along the zero effect line. For site codes, see the legend to Figure 3.

frequency and intensity of disturbance events, such as fire and insect herbivory, which lead to tree establishment and mortality. The natural disturbance regime at the seven western FFS sites incorporates a similar frequent, low-severity fire regime, characterized by surface fires that cause shifts between multimodal and unimodal diameter distributions as new cohorts establish and are subsequently eliminated. The frequency and severity of surface fire affects the variability in size of tree killed, and past fire exclusion and suppression have contributed to multimodal, decreasing or reverse J, or irregular distributions.

Diameter distributions are considered here in the context of overall changes in forest structure associated with fuel reduction and forest restoration treatments. Although objectives for some common metrics of fuel reduction, such as mass of woody fuels, may easily be obtained with treatment, objectives for other metrics, such as diameter distributions, are more difficult to obtain because both tree number and size are involved. Fuel reduction and forest restoration treatments may leave small numbers of trees in the small diameter classes in the short term, but the treatment cannot create more trees in larger size classes. Diameter distributions are used here as one component of stand-level forest structure, in addition to metrics such as overstory canopy cover, basal area, stand density index, mean tree height, and crown base height.

Tree diameter is a ubiquitous measure of tree size and is easily but rarely presented as a stand-level forest structure attribute by displaying the frequency of measurements across some range of size classes. More often it is summarized to a single metric of stand-level forest structure as the mean or quadratic mean diameter and by the number of trees above some threshold value of diameter. The SD of all the measured tree diameters is a measure of the variability in tree size and may be indicative of diversity within the stand (McElhinny et al. 2005). Others have applied the Shannon-Weiner index to diameter measurements for a tree size diversity index when comparing between diameter distributions (Wikstrom and Eriksson 2000). The Kolmogorov-Smirnov two-sample test, from the family of χ^2 tests, is useful for comparing the similarity of two ordered distributions and can be used to assess how well the cumulative distribution of sample data conforms to that of a reference or theoretical distribution (Yang et al. 2004). Other tests include the Kullback-Leibler criterion and the Gini coefficient; both these tests, like the Kolmogorov-Smirnov two-sample test, may suggest a difference between a reference or pretreatment distribution and a test or posttreatment distribution but fail to indicate the magnitude and direction of any difference between the two distributions (Menning et al. 2007). The Menning departure index is reported to measure the magnitude and direction of a shift between a reference or pretreatment distribution and a test or posttreatment distribution and is insensitive to the number of bins or diameter size classes (Menning et al. 2007), yet it can be shown that this index produces results nearly identical to those of the mean diameter.

In addition to fire regime, the seven western FFS study sites have a number of common features that include a history of fire exclusion and suppression, timber harvest (except at Southern Sierra Nevada), dominance of long-lived fire-resistant conifers (especially ponderosa pine), and accumulations of both woody and live fuels (particularly high numbers of small trees that serve as ladders for fire into the canopy). Although unique reference conditions were not developed for each western FFS study site, reference conditions for the ecosystems represented by the FFS study sites were generally available (Arno et al. 1997, Fulé et al. 1997, Harrod et al. 1999, Youngblood et al. 2004) and provided descriptions of diameter distributions that aided in prescription development at each site. Reference conditions for dry ponderosa pine late-successional forests in portions of the Cascades, for example, are relatively broad unimodal distributions that contain less than 10 trees ha^{-1} in each 10-cm diameter size class with the mean diameter about 50 cm (Harrod et al. 1999, Youngblood et al. 2004). In contrast, long-lived fire-resistant conifers, accumulations of woody and live fuels, and similar reference conditions are not shared across five eastern FFS study sites and, thus, as previously indicated, were excluded from this work.

These results provide evidence of similar responses to treatments across western FFS study sites. Thin treatments, especially at Northeastern Cascades, Southern Cascades, and Central Sierra Nevada sites, resulted in posttreatment diameter distributions to be right-shifted because small and medium diameter trees were cut. At the six sites where thinning was applied, unique operational standards were used to guide commercial thinning. Thin treatments by design focused on the smaller diameter classes, yet the smallest size classes may have been below utilization standards at some sites, which resulted in the retention of high numbers of trees in the smallest size classes at some sites. Differences between thin and burn treatments at the Central Sierra Nevada site probably would have been greater had the thin treatment not included mechanical mastication of small diameter saplings. The thin treatment at the Northeastern Cascades sites may have been constrained by the use of helicopters for yarding. Effects of burn treatments were more variable than those of thin treatments, yet this treatment caused posttreatment diameter distributions to be right-shifted because small diameter trees were killed and the number of large trees was left largely unchanged. Tree species at all seven western FFS study sites generally increase in fire resistance with increasing diameter, especially ponderosa and Jeffrey pines and Douglas-fir (Sieg et al. 2006, Hood et al. 2007). Predicting mortality of large trees after burning is more problematic and probably involves a combination of secondary factors, particularly bark beetle attacks (Parker et al. 2006, Youngblood et al. 2009), in addition to primary effects of fire. Thin + burn treatments caused posttreatment diameter distributions to be right-shifted, especially at the Blue Mountains, Northern Rocky Mountains, Southern Cascades, and Central Sierra Nevada sites, and the shift tended to be larger than that with either thin or burn treatments. The effect appears to be additive; the number of lives trees in small and medium sized diameter classes were reduced by thinning, and the numbers of live trees in the smallest diameter classes were further reduced by the subsequent burning. Thus, more of the total diameter distribution was affected by the thin + burn treatments than by either the thin or the burn treatment.

Shifts in diameter distributions toward late-successional structure as a result of the FFS study fuel reduction and forest restoration treatments are viewed here as an essential component of restoring resiliency in low-elevation dry forests. The overall effect size of the thin, burn, and thin + burn treatments indicates that treatments increased the cumulative frequency of large diameter trees and moved diameters from multimodal, decreasing or reverse J, and irregular distributions toward unimodal distributions. This shift toward more unimodal distributions probably has distinct implications for managing with climate change uncertainty. Active treatments may be steps in creating structures that are better able to resist the influence of climate change and are resilient to change and enable forests to respond to change (Millar et al. 2007). Low-elevation dry conifer forests of the western United States characterized by unimodal diameter distributions that emphasize large diameters probably are better able to resist the undesirable or extreme effects of fires (Youngblood et al. 2004). Stands with this structure, when dispersed across larger landscapes, may provide landscape-scale resiliency because they are better able to return to prior conditions after a disturbance whereas stands with other distributions may experience greater severity in disturbance during the same event. Stands with this structure also may respond favorably to climate change because they have been realigned to incorporate both current and expected natural disturbance processes.

Work reported here is evidence that fuel reduction and forest restoration treatments, conducted in the context of restoring both forest structure and processes, resulted in detectable changes in tree diameter distributions. These findings provide a necessary foundation for future and ongoing assessments of vegetation dynamics, treatment economics, social valuation, carbon sequestration, and interactions among treatments. Further work with potentially more severe treatments, such as greater basal area reduction or greater mortality from burning, would aid in quantifying ecosystem resilience. When extended beyond this first set of treatments, this work provides a foundation for assessing the effects of continued or repeated fuel reduction and forest restoration treatments for reducing fuels and accelerating the development of late-successional stand structure in low elevation dry coniferous forests across the western United States. As part of the national Fire and Fire Surrogate study, this work identifies the nature and strength of ecological responses that cross ecosystem boundaries and provides managers a framework for predicting the outcome of fuel reduction and forest restoration treatments.

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